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Ecological Criteria for Evaluating Wetlands

by

Ronald G. Osborn

Midcontinent Ecological Science Center

National Biological Service

4512 McMurry Avenue

Fort Collins, CO 80525-3400

ABSTRACT

Wetland evaluation methods historically examine the functions a wetland performs and the value to society the functions provide. Wetland evaluation methods vary more in their procedures for measuring and scoring functions than in selection of functions used. Ecological and conservation evaluation methods focus on ecosystem qualities rather than societal good and are used to identify natural areas for conservation. Criteria used in 14 wetland evaluation methods are examined and compared to criteria used in ecological evaluation methods. Represented criteria include diversity, size (or area), landscape pattern, rarity, productivity, importance to wildlife, representativeness, naturalness and ecological integrity, and ecological fragility (or replaceability and threat). It is recommended that ecological criteria for permit applications be selected to address regional conservation priorities and associated information at regional scales, and use landscape patterns and geographic information tools to effectively interpret those patterns. While geographic information systems can help formalize criteria for recordkeeping and long-term monitoring, improved agency collaboration and data sharing will be necessary to realize their potential benefits. Finally, particular methods of scoring criteria and comparing sites are less important than the selection of criteria and their measurement. Nevertheless, it is recommended that individual criteria or wetland function measures be compared rather than combining measures into a single score for comparison.

Keywords: wetland evaluation methods; ecological criteria; section 10/404 permit application and review.

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PREFACE

The work was performed at the National Biological Service's Midcontinent Ecological Science Center (MESC) in Fort Collins, CO. This report was prepared by Ronald G. Osborn in MESC's Landscape and Habitat Analysis Section. Other participants in the work at MESC included Chuck Solomon and Natalie Sexton. Richard Schroeder, Adrian Farmer, and Greg Auble at MESC helped in the formulation of many ideas expressed in this report. Other reviewers providing valuable assistance were James Roelle, MESC; Rick McCorkle, U.S. Fish and Wildlife Service's Delaware Estuary Program; and Carl Brown, retired.

Introduction

Wetlands constitute only 5% of the surface area of the conterminous United States (Dahl 1990), yet they provide important habitat for about 33% of the plants and animals listed as threatened or endangered and are essential nesting, stop-over, and wintering areas for more than 50% of the nation's migratory birds (U.S. Fish and Wildlife Service [USFWS] 1990b). The view of wetlands as a national resource advanced greatly in the mid-1970s with a variety of major initiatives (see Kusler 1979) including the Clean Water Act. Subsequently, the administrative need for wetland evaluations became widespread and a variety of approaches were identified (Lonard et al. 1981). At present no method is predominant and many evaluation methods are in use.

Wetland evaluation methods analyze the functions a wetland performs and the relative value those functions provide to society (Adamus and Stockwell 1983, Bond et al. 1992). For example, wetlands function to help lower flood peaks and to provide wildlife habitat. Related values to society include reduction in property damage and opportunities for recreation. In general, wetland evaluation methods rate areas according to their attributes (e.g., duration and timing of surface inundation, or the wetland-dependent avian species that breed there). Criteria express a site's attributes and measure a wetland's functions *per se*, not necessarily the function's value to society. The methods typically do not consider project activities or impacts on wetlands, although methods may be part of an environmental assessment or regulatory activity.

Evaluation methods are also used to identify natural areas for conservation or other purposes (Usher 1986). Similar to wetland evaluations, van der Ploeg and Vlijm (1978) describe ecological evaluations as assessments of ecosystem qualities *per se* or socio-economic procedures to estimate the value of the natural environment to society. Typically, both wetland and ecological evaluations use criteria and attach non-economic values to sites where the word "best" implies making both value judgments and comparisons (Usher 1986). Such evaluations are used for environmental impact assessment, land-use planning, planning systems for protected areas, or management planning of individual protected areas (Smith and Theberge 1986).

The purpose of this paper is to: 1) compare ecological criteria used in the context of ecosystem quality and societal good, and 2) discuss their use in 14 wetland evaluation methods listed in Table 1. Details of the evaluation procedures are of lesser interest, but it is necessary to understand the nature and trends in wetland evaluation studies to more fully appreciate the role criteria play. For example, it became clear that while two methods (Adamus 1983, Adamus et al. 1987) differed greatly in the mechanics of scoring and measurement of specific variables, the criteria themselves remained essentially the same and in this review have been summarized as a combined method in Tables 2 and 3. Recommendations are given concerning potential use of criteria and potential data sources applicable to rapid assessment for the U.S. Army Corps of Engineers (USACOE) 404 regulatory program.

The Development of Wetland Evaluation Methods

Much has been written on the functions and values of wetlands (Greeson et al. 1979, Sather and Smith 1984). The USFWS has created a wetland values database containing abstracts of over 2,000 articles on the subject (Stuber 1983). One of the most studied functions of wetlands is habitat for fish and wildlife (Lonard et al. 1981), yet there remains concern about the inadequate literature base regarding its evaluation (Stuber and Sather 1984).

Table 1 summarizes 14 wetland evaluation methods reviewed for their use of ecological criteria. Procedures for these wetland evaluations vary from general guidelines for local site comparisons (Ammann and Stone 1990) to comprehensive assessments of entire landscapes (Gosselink and Lee 1989). Data needs to support the methods vary from interpretation of aerial photography and simple site inspection to long-term historical trend analysis and detailed information on population abundance and distribution. Time requirements vary from less than a day to months or years. In general, wetland evaluation methods are intended to be easily applied in a consistent manner by non-technical personnel at either a local (Ammann and Stone 1991, Municipality of Anchorage 1991, Golet 1976, Hollands and McGee 1986), regional (North Carolina Department of Environment, Health, and Natural Resources [NCDEHNR] 1995, Ontario Ministry of Natural Resources and Environment Canada [OMNREC] 1984, USACOE 1988), or national (Adamus 1983, Bond et al. 1992) scale. However, the scope of functions included and measurement of variables used as indicators of those functions are still too time consuming for widespread use in the 404 Regulatory Program. For example, even simple methods may take eight hours to apply and involve a field trip to a wetland (USACOE 1988). The challenge has been that models must be bounded by complexity that makes them useless and simplicity that makes them trivial (Starfield 1990).

The 14 methods examined in this report generally use comparable wetland functions and similar criteria for evaluating habitat for fish and wildlife. Most methods (Table 2) address diversity, productivity, and abundance of wildlife and their habitats. Only a few of these wetland evaluation methods (Cable et al. 1989, Durham et al. 1988, Gosselink and Lee 1989) do not include a wildlife habitat component based on some type of wetland classification adapted from either Golet (1976) or Adamus and Stockwell (1983).

Particular measures for ecological and habitat criteria have generally relied on one or more classifications of wetlands and associated subjective scores for a site's particular characteristics. In this sense the classification itself becomes an indicator of value. Methods may represent only minor differences by modifying a classification for a particular region, or modifying scores related to the classification or individual measure. Two methods (Adamus 1983, Adamus et al. 1987) differed greatly in the mechanics of scoring and measurement of specific variables, but the criteria themselves remained essentially the same and in this review have been summarized as a combined method in Tables 2 and 3.

Standardizing criteria in wetland evaluation methods does not necessarily make the evaluation process objective. Fuller and Langslow (1986) noted that quantification of criteria does not imply objectivity since the choice of attributes may still be subjective. As stated earlier, the idea of "best" implies that both value judgments and comparisons have been made (Usher 1986). Nevertheless, evaluations have become well established, and the use of standard methods and guidelines for their application can provide both improved consistency and reduced subjectivity (Glooschenko et al. 1988).

Techniques of scoring

All methods examined deal with scoring variables related to multiple objectives. Scores for individual measures may be summed into a cumulative score for a wetland function

(USACOE 1988, Municipality of Anchorage 1991), or totaled for all wetland functions (Golet 1976, Hollands and McGee 1985, NCDEHNR 1995). Some methods emphasize individual criteria whether by applying different weights to individual indicator values (Golet 1976, USACOE 1988) or by scaling indicator values differently which is the same as implied weights (OMNREC 1984). A few methods are analogous to the Habitat Evaluation Procedures (HEP), developed for individual species (USFWS 1980), and use an index value of quality multiplied by area to compute a wetland score. For example, Ammann and Stone (1991) refer to the quality as a functional value index and the score as wetland value units; Durham et al. (1988) refer to a habitat quality index and habitat unit values.

Several methods (Adamus 1983, Adamus et al. 1987) do not use numerical values *per se*, but use answers to interpretive keys to estimate the likelihood that a function may exist or occur as high, moderate, or low. Adamus (1983) rates the function’s likelihood as high, moderate, or low for opportunity, effectiveness, and significance. Opportunity considers whether a wetland has a chance to fulfill a particular function. Effectiveness considers the degree to which a function is fulfilled. Significance considers the degree to which the performed function is valued by society. The final functional significance value is the interpreted result of the interactions of opportunity, effectiveness, and significance, rather than a mathematical formulation of conditional probabilities. The method uses many predictors (n=75) and questions (n=153) with simple yes/no answers. It can be applied even when data for preferred measures are unavailable, but Odum et al. (1986) found that omitting information tended to generate "artificially" moderate wetland values.

There is some evidence that weighting individual measures changes the final score little (USACOE 1988). This could be an artifact of the number of measures; with a large number of measures, a dramatic change in one measure has only a limited effect on the overall score (Odum et al. 1976). In practice, the rationale for adjusting weights and scaling measures seems to stress discrimination among wetlands rather than whether differences are meaningful (Hollands and McGee 1985, Gersib et al. 1989). Correlation among similar measures, such as Durham et al. (1988) found for variables in their community model, poses another problem. Two criticisms are common to all methods—failure to address uncertainty of the measurement values themselves (Odum et al. 1976) and combining criteria into a final score without the use of utility theory techniques (Smith and Theberge 1987). Tools like the multi-attribute tradeoff system (Brown and Valenti 1983) could provide a means to incorporate utility theory in these methods. At present, familiarity in applying the methods must be gained before the sensitivity of outcomes to particular values can be anticipated (Odum et al. 1986).

Another trait shared by many wetland evaluation methods is special criteria, levels, or procedures that may supersede or override other considerations. Larson (1976) calls these "outstanding" natural or cultural values. Ammann and Stone (1991) call these "noteworthiness" values. Several methods (OMNREC 1984, USACOE 1988) identify "special features" to alert the evaluator to what they view as a "red flag" index. Some indices may be more administrative than ecological. For example, there are often laws protecting wetlands or their resources that require coordination with other agencies. Criteria related to these laws can alter evaluators, showing when appropriate agencies must be contacted (USACOE 1988). Municipality of Anchorage (1991) identifies "red flag" species that are rare or sensitive to the size or condition of a wetland and elevate the site's value when present. These criteria may be measured directly rather than using indicators. However, they are not criteria *per se,* since they are often related to other ecological criteria (e.g., rarity or abundance of wildlife or communities).

A few methods (Larson 1976, Adamus 1983, and Bond et al. 1992) use distinct steps in their evaluations where information and time requirements are more demanding with each step. Wetlands with "easily identifiable and widely acknowledged values" can be identified quickly and early, saving time and reducing the amount of information that would otherwise be evaluated (Larson 1976). Adamus (1983) uses a Comparative Analysis to discriminate among wetlands scoring the same in the initial Threshold Analysis. Sather and Stuber (1984) further suggest that "red flag" factors be added to the Adamus method so that there would be no need to go on with the procedure if sufficient "red flag" values exist. Bond et al. (1992) eliminate degraded sites, with little chance of restoration early in the process, when project benefits are great. Such approaches imply the use of a screening procedure, or coarse filter, where the most significant criteria are applied first. Those sites that pass through or cannot be discriminated with the coarse filter must then be evaluated with additional criteria or methods.

Only one wetland evaluation method(Gosselink and Lee 1989) does not focus on site evaluations of either individual wetlands or individual projects. Nevertheless, a few site-based or project-based evaluation methods include landscape features that influence the evaluation of a site (e.g., adjacent cover types or land use, hydrologic connectedness, or proximity to nearby wetlands [Adamus 1983, Golet 1976, USACOE 1988, Municipality of Anchorage 1991, Ammann and Stone 1991]). Regional landscape features may also be included. One approach (OMNREC 1984) emphasizes the attributes of a site as measured against a standard for a region (e.g., contribution to regional diversity, representativeness or loss of a wetland type). Another approach (Gosselink and Lee 1989) considers regional patterns that may actually influence the values of a site (e.g., habitat size, contiguity, and isolation).

Ecological Criteria in Evaluations of Natural Areas and Wetlands

Criteria are used to express a site's attributes in a form that can be used in an evaluation (Usher 1986). Measurement criteria range from assigning scores based on subjective preferences to measuring environmental variables directly (Smith and Theberge 1987). Measurements may be relatively simple, such as phosphorous concentration (Gosselink and Lee 1989), or involve complex indices requiring knowledge about the distribution and abundance of many taxa at multiple scales (Klopatek et al. 1981). Cooper et al. (1990) indicate that criteria used to assess wetland functions may be measured directly (e.g., large numbers of breeding waterfowl), deduced from related site data (e.g., suitable characteristics for secure nesting sites and abundant food supply are present), or inferred from the site's membership in a particular category (e.g., a palustrine emergent marsh).

Margules and Usher (1981) defined a criterion as the "basic scientific concept, and not the value placed by society on the concept," and reviewed criteria for assessing wildlife conservation potential from nine studies. Criteria from eight additional studies were examined by Margules (1981) [cited in Usher 1986:13]. The studies they reviewed were primarily on a local or regional scale in human-dominated landscapes. Smith and Theberge (1986) examined criteria from 13 additional studies including evaluations of wetland, freshwater, and marine natural areas. The scale of these studies ranged from local to international and included both highly industrialized and wilderness areas. And ecological criteria from 14 wetland evaluation methods are reviewed here (Table 1).

Each evaluation scheme has its own set of criteria defined in a different way. The names for similar criteria and how they are measured vary from system to system. Margules and Usher (1981) attempted to classify similar criteria into generic classes. Smith and Theberge (1986) used similar classes of criteria in their review. A similar classification of ecological criteria is used here in review of wetland evaluation methods (Table 2); comparisons with earlier reviews of ecological evaluation methods are presented in Table 3.

Diversity

A generally accepted definition of diversity is the number of elements and their relative abundance (Smith and Theberge 1986). The elements most frequently of interest are assemblages of species for taxa such as birds. Abundance is typically the number of individuals, but can be measured using biomass or even trophic connections in food webs (Goodman 1975). In addition to species, elements of diversity may be habitats or communities, provided a classification exists so the different types may be measured (Pielou 1977). Wetland classifications are varied and may include characteristics of life forms, flora, water regime, water chemistry, and geomorphology (see Cowardin et al. 1979, Novitzki 1979, and Kangas 1990).

The Shannon-Weaver (1964) index (***H***) and Simpson (1949) index (***D***) are two common measures of diversity:

$$\boldsymbol{H} = -\sum_{i=1}^{S} \boldsymbol{p_i} \ln(\boldsymbol{p_i}) \; ; \qquad \mathbf{D} = \sum_{i=1}^{S} \boldsymbol{p_i}^2$$

where $\boldsymbol{p_i}$ is the proportion of the ***i***th species in a sample. ***H*** has a minimum value of zero for a monoculture community and a maximum for a sample when elements are equally abundant. That is, higher values represent either more species or more even representation of species. ***D*** has the property of being approximately equal to the probability that two individuals drawn at random from a sample are of the same species. Hence, ***D*** has a maximum value of one for a monoculture with smaller values representing higher diversity. Both these measures of diversity, and others, can be mathematically specified by a single equation from which the different measures are derived by varying a single parameter (Hill 1973). The different indices vary primarily in how much weight is given to common and rare species (Hill 1973).

Species richness, the number of species in a sample or community, is often used to measure diversity because it is more easily determined. But as sample size increases, the number of species tends to increase (Usher 1986), whether the sample is a collection of individual organisms or a geographic area (Preston 1962, Simberloff 1978). Techniques are available to correct for the number of individuals in a sample (Simberloff 1978), and the effect of area may be factored out with regression techniques (Connor and McCoy 1979). Dony and Denholm (1985) use an assessment index for diversity based on the ratio of the observed value to the predicted value from a species-area relationship, but frequently area is treated as a separate criterion (see Area below). van der Ploeg and Vlijm (1978) stress that unless the effect of area on diversity is removed, the criterion is completely unreliable for objective comparison between sites.

Whittaker (1972) related the notion of diversity to geographic scale and spatial context by introducing the concepts of alpha, beta, and gamma diversity. Alpha diversity is defined as the number of species in a particular site. [For birds this is the diversity in a small area of a few hectares or less with a uniform vegetation structure (Karr 1976)]. Beta diversity reflects the change in species composition along an environmental gradient or series of habitats. [For birds the gradient of interest is frequently vegetation complexity or height (Cody 1975)]. The total geographic diversity within a large region is gamma diversity. Unfortunately, measurement of beta diversity (***BD***) is labor intensive even when using the relatively simple index proposed by Whittaker (1972):

$$BD = Sc / \overline{S}$$

where ***Sc*** = the number of species in a composite of several alpha sites, and **S** is the mean number of species in alpha sites.

The rationale for using diversity is not frequently stated. The presumed connection between diversity and stability is refutable (see Margules and Usher 1981). In wetland evaluations it is argued that areas of high vegetation diversity, in general, provide interspersion of habitats for more species of wildlife. Studies of birds in structurally simple marshes (Weller and Spatcher 1965) are typically the only reference to such arguments. Correlation between diversity of different taxa (e.g., plants and invertebrates, vertebrates and invertebrates, birds and mammals) are not generally well founded (Oliver and Beattie 1993, Learner et al. 1990) and may be more a consequence of beta diversity or area (see Area below). On a local scale, insular area *but not* vegetation diversity of mixed oak patches of varying size was the significant factor in predicting the number of breeding bird species (Galli et al. 1976).

Succession has an important influence on diversity, with mature sites typically having higher diversity (Mannan 1982, cited in Harris 1984:65) and higher abundance of individual species (Hamel 1989). This may be confused with increased beta diversity caused by natural disturbances. Riverine channels maintain portions of the floodplain in early seral vegetation and enhance diversity in plant communities and structural composition contributing to diverse wildlife communities (Cooper et al. 1990).

Many wetland evaluation methods (Golet 1976, Hollands and McGee 1986, Ammann and Stone 1991, Adamus 1983, Adamus et al. 1987, Municipality of Anchorage 1991) emphasize a maximum local diversity philosophy and use the number of wetland classes (life forms) and other structural measures of edge as indicators of wildlife diversity (see Golet 1976). Noss (1983) summarizes the shortcomings of this philosophy where edge-adapted species are mostly wide-spread species in urban and agricultural landscapes. Too often emphasis is placed on maintaining high species diversity instead of characteristic "native diversity" (Noss and Harris 1986). Local species diversity can be increased by habitat fragmentation, but added species are typically common species in no danger of becoming rare, and regional diversity remains the same (Noss 1983). Murphy (1989) emphasizes that including non-native species and study plots that are not relevant to the geographic scale important to conservation concerns, only results in confusion over self-evident truths of conservation biology that big reserves are good, small ones not so good, and that some species need our attentions while others can take care of themselves.

Others have addressed these concerns by excluding hybrids, naturalized, and introduced species from their measure of richness (Dony and Denholm 1985), or by factoring species quality with species richness, and assigning higher values to sites containing species with smaller breeding populations statewide (Cable et al. 1989; see also Rarity below).

A number of studies imply that diversity indices confuse interpretations of an evaluation. Tramer (1969) and Kricher (1972) found that changes in bird species diversity were closely correlated to species richness while the relative abundance component remained stable. Morris and Lakhani (1979) indicated that the Shannon-Weaver index was more sensitive to differences between sites than the Simpson index. Gotmark et al. (1986) used species richness, rarity, and abundance to rank 15 wet meadows and 47 bogs in Sweden and found that the Shannon-Weaver and Simpson indices corresponded poorly with the authors' subjective rank.

Diversity, as a measure by itself, is of questionable value since its interpretation often requires examination of the species complement, environmental conditions, and site history (Dony and Denholm 1985). The focus on numbers of species tends to obscure the fundamental point that protecting intact ecosystems is what is at stake (Noss 1983). Knowledge of the species complement may be the most relevant attribute in a wetland evaluation.

Area

Usher (1986) notes three aspects of area that concern conservationists, especially in relation to evaluation of a site: 1) the relationship between the number of species and area, 2) the concept of minimum viable population size, and 3) the related idea of minimal critical ecosystem size (i.e., is there some size below which the community cannot function [Lovejoy and Oren 1981] and is not worth conserving?).

The theoretical relationship between number of species and size of the area examined has long been known (Arrhenius 1921, Preston 1962) and is expressed as:

$$\mathbf{S = c\,A^{z}}$$

where **S** is the number of species, **A** is area, **c** a constant that varies with taxon and geographic region, and **z** is a constant measuring the slope of the line relating log (**S**) and log (**A**). Connor and McCoy (1979) found this model to provide the best fit in a large number of data sets, and Boecklen and Gotelli (1984) found that about half the variation in the species richness of these data sets was explained by the species-area curve (see Figure 1). Kilburn (1966) indicated that the relationship can be used to determine the minimum area of a community and predict the number of species in an area larger than those sampled.

Size is important to nature reserves from the standpoint of capturing and maintaining the diversity of species and genes in a region (Smith and Theberge 1986). Principles derived from island biogeography (MacArthur and Wilson 1967) show that species richness reaches an equilibrium between immigration and extinction dependent on island size and isolation (Figure 2). MacArthur and Wilson (1967) also suggested the island analogy for continuous natural habitats that have been fragmented into habitat "islands." Diamond (1975) extended the analogy to provide simple guidelines for design of nature reserves (see Landscape Pattern below).

Patches of forest or other natural habitat in developed urban and agricultural landscapes can be considered ecologically similar to islands (Harris 1984, Diamond 1975). Interior habitats for which a species-area relationship has been demonstrated include isolated forests (Galli et al. 1976), prairies (Samson 1980), mountain ranges (Picton 1979), and wetlands (Tyser 1983). Taxa for which the relationship was demonstrated include birds (Blake and Karr 1987), mammals (Lomolino 1982), reptiles (Jones et al. 1985), amphibians (Laan and Verboom 1990), invertebrates (Murphy and Wilcox 1986), and plants (Wade and Thompson 1991).

More recently, studies have indicated that composition of wildlife communities varies with habitat size in a nested distribution of species (Patterson and Atmar 1986, Blake 1991). In particular, species that occur in smaller habitat patches are a subset of the species that occur in larger patches and large patches tend to be favored by habitat specialists (Blake and Karr 1987).

Brown and Dinsmore (1986) looked at 30 seasonal and semipermanent marshes from 0.2 to 182 ha in size and found that area explained the greatest amount of variability in species richness of the marshes. Not all marsh species are area sensitive (e.g., common grackle [*Quiscalus quiscula*], yellowheaded blackbird [*Xanthocephalus xanthocephalus*], redwing blackbird [*Agelaius phoeniceus*]). Robbins et al. (1989) found that 50% of the breeding forest birds in mid-Atlantic states were habitat size-dependent, as did Galli et al. (1976) in mixed oak patches (dominated by *Quercus alba*, *Q. velutina*, and *Q. borealis*). The critical size of wetlands in Iowa where whole associations of species have been lost was about 5 ha (Brown and Dinsmore 1986).

Much attention has been focused on the issue of whether a single large or several small reserves are best for conservation (see Simberloff and Abele 1976 and subsequent replies), but the important issue for conservation is to identify the species composition of these communities and evaluate this information in relation to conservation goals (Schroeder and Keller 1990). If species that only occur in the large areas are of concern from a conservation point of view, then area must be considered. Two groups of species that have received the most attention in this regard are large mammalian carnivores and area-sensitive or forest interior birds (Harris 1988, Robbins et al. 1989; see also Landscape Pattern below). Habitat size-dependent eastern forest species are primarily neotropical migrants, nest on or near the ground in forest interiors, and raise a single brood from a small clutch (Samson 1980).

A second rationale for the importance of area is that different species have different range requirements and minimum viable population sizes (Shaffer 1981, Gilpin and Soule 1986). Shaffer (1981) advocates a focus on species at the top of the food web, reasoning that if we are successful in providing sufficient room for their survival, other species that require less space should also survive. Zeveloff (1983) estimated that 40,000 ha per reserve are needed to maintain a viable black bear (*Urus americanus*) population. Harris (1988) suggested that the Florida panther's (*Felis concolor coryi*) home range of 50,000 ha, where males can be lethally territorial, might require more area for a viable population than exists in any single tract.

Although area is used more than any other criterion in wetland evaluations (Table 2), it is often an ancillary indicator for diversity and productivity without regard to particular species needs (USACOE 1988). Larson (1976) considers large wetlands that dominate the landscape as sufficient rationale for preservation. Golet (1976) considers wetlands larger than 200 ha sufficiently rare to be of statewide or regional significance in the glaciated northeast. Other methods (Ammann and Stone 1991; Missouri Department of Conservation and U.S. Soil Conservation Service 1990, cited in Schroeder and Haire

1993) use area over too small a range (< 2 ha) to adequately reflect the influence of wetland size on species richness.

Of 14 components in its wildlife habitat potential function, the Municipality of Anchorage (1991) considers wetland area the most important, accounting for up to 40% of the possible function score. The Municipality of Anchorage (1991) also uses area-sensitive species (Northern Harrier, Hudsonian Godwit, and others) as indicators of high-value wetlands. Modified area relationships are used based on the other component values to assess habitat quality where a small high-quality wetland is considered equivalent to a large low-quality wetland (OMNREC 1984, Municipality of Anchorage 1991). [It should be noted that the species-area relationship also implies that a large high-quality wetland contains greater biodiversity than any number and combination of small high-quality and large low-quality sites of similar type]. Dony and Denholm (1985) use the ratio of observed species richness to that predicted from the species-area relationship to assess habitat quality.

Cable et al. (1989) use the species-area relationship based on an optimum size (i.e., a small tract is ecologically inferior and a large one may be economically disadvantageous) of 40 ha for palustrine wetlands and 60 ha for emergent estuarine wetlands. Samson (1980) describes a minimum area point on the species-area curve where a 5% increase in number of species requires a doubling in habitat size. Robbins (1979) suggests that the species requiring the largest minimum area be used to determine the critical habitat size of interest. Both measures appear greater than Cable's optimal size, perhaps 40 ha and greater than 100 ha, respectively, for a mixed oak forest (Galli et al. 1976, Samson 1980), and several hundred to a thousand ha for bottomland forests (Graber and Graber 1976). In mid-Atlantic states, Robbins et al.

(1989) estimate there is less than a 50% chance of finding 10 of 26 area-sensitive forest birds in forest patches less than 150 ha in size. Habitat patches necessary for supporting viable populations of certain birds may need to be larger than expected based on surveys of singing males alone (Gibbs and Faaborg 1990).

Gosselink and Lee (1989) do not use a species-area relationship *per se* but instead rely on the distribution of patch size and its comparison to pre-development conditions. Their approach is more similar to naturalness or representativeness criteria (see below).

In summary, the species-area relationships range broadly among different wetland types and taxa. It appears most important in forested wetlands where large blocks of contiguous habitat are needed to support a full complement of avifauna, especially neotropical migrants.

Landscape pattern

Noss and Harris (1986) describe a landscape as a heterogeneous and ever-changing entity that nevertheless maintains a constancy and predictability of disturbance and recovery patterns in a time scale meaningful to human beings. It is also the aspect of heterogeneity in a landscape that provides "edge," (defined as the place where plant communities meet or where successional stages or vegetative conditions within plant communities come together [Thomas et al. 1979]). Where habitat edges exist, an ecotonal community develops, containing many of the species of the overlapping communities as well as edge-adapted species that either primarily occur or are most abundant there (Odum 1971). Thus, edge effect and the related phenomenon of habitat fragmentation are important issues when discussing landscape pattern.

Edge effect has been a fundamental principle of wildlife management and is a major factor encouraging a maximum local diversity philosophy (Noss 1983). Edge has high cover density (Johnson et al. 1979) and food availability in accordance with high primary productivity (Ranney et al. 1981). Edge-adapted species are often habitat generalists characteristic of disturbed environments (Harris 1988). Game animals are commonly edge-adapted, as are animals of agricultural, suburban, and urban landscapes (Whitcomb et al. 1976). Vertical distribution of foliage within a habitat and interspersion of vegetation types are correlated with the number of resident bird species (MacArthur and MacArthur 1961, Roth 1976). The influence of the proportion and interspersion of open water, vegetation structure, and plant communities on diversity and production of birds in a marsh (Weller and Spatcher 1965) has caused these measures to be the most frequently included in wetland evaluation. (Only Cable et al. [1989] completely omitted them, although they are implied by phrases like "good cover and food for a diversity of wildlife" [NCDEHNR 1995] in other methods.) Durham et al. (1988) use nine variables, mostly related to vegetation composition and structure, to measure site-specific habitat quality in their avian community model of bottomland forests. Most measures of edge in wetland evaluations, however, are targeted at waterfowl abundance and production.

Samson and Knopf (1982) have questioned the logic of management for maximum local diversity benefiting widespread and opportunistic species over more sensitive and rare species, especially for nongame birds. Noss and Harris (1986) suggest that too much emphasis is placed on site content (what is on and characteristics of a site) and too little on its context (how it is part of the landscape), since surrounding land can have significant effects on species composition and diversity within a habitat island (Harris 1984) or adjacent wetland (Leidy et al. 1992).

The process of fragmentation and insularization of native habitats in an expanding urban and agricultural landscape is one of the main concerns of conservation biology (Harris 1984). Effects of habitat fragmentation extend beyond species-area relationships already discussed and may lead to equally or even more serious indirect effects than the direct loss of habitat (Whitcomb et al. 1981, Wilcove et al. 1986). Consequences of fragmentation include: 1) reduced colonization of habitats with increased isolation and barriers to movement and dispersal; 2) loss of habitat heterogeneity; 3) detrimental interactions between habitat components; 4) edge effects that further reduce a fragment's effective area; and 5) secondary effects from new or disrupted ecological interactions. All these factors produce smaller, more isolated populations. Habitat fragmentation may be the most serious threat to biological diversity and the primary cause of the present day extinction crisis (Wilcox and Murphy 1985).

Robbins et al. (1989) note that proximity of other forests appears to enhance the effective area of an isolated patch in terms of bird species richness. Whitcomb et al. (1981) used a gravity model for assessing isolation of forest fragments within 3 km of a forest's edge, and found the measure was significantly correlated to number of interior bird species in forest patches. Brown and Dinsmore (1986) demonstrated the importance of isolation in Iowa marshes. They found that some area-sensitive species (black tern [*Chlidonias niger*], redhead [*Aythya americana*], Canada goose [*Branta canadensis*], and swamp sparrow [*Melospiza georgiana*]) occurred in smaller marshes when these marshes were within wetland complexes (expressed as ha of marsh within

5 km). Proximity to wetlands is used in many evaluations. Wetlands interconnected by surface water are often deemed more important because the water provides movement corridors between more widely distributed sites (USACOE 1988). The threshold distance for considering adjacent wetlands varies from 4.8 km (Ammann and Stone 1991, USACOE 1988 for forested regions) to 1.5 km (Golet 1976, OMNREC 1984). For prairie regions, the USACOE (1988) considers wetlands to be isolated if they are not part of a complex (i.e., where the third closest wetland is greater than 0.8 km away).

Structurally different vegetation at the edge of forest fragments extends 10-30 m inward (Ranney et al. 1981). Narrow corridors through forests are frequently colonized by avian brood parasites (e.g., cowbird), nest predators (e.g., blue jay, American crow, and common grackle), and nonnative nest hole competitors (e.g., starling) (Ambuel and Temple 1983). Their effect may extend from 300-600 m into the forest (Wilcove 1985), effectively leaving no true forest interior in circular patches smaller than 100 ha. Even larger tracts of linear bottomland forests provide no suitable habitat for interior forest species without being contiguous to forested uplands (Gosselink and Lee 1989).

Terrestrial mammals are highly vulnerable to barriers such as roads between habitats they use (Mader 1984) and, among these, carnivorous mammals are probably the group most vulnerable to fragmentation (Harris 1988). Noss and Harris (1986) suggest that riparian strips fulfill the function of allowing species to move between preserves and other natural areas. However, Wilcove et al. (1986) suggest that adjacent land use that allows target species to exist at least marginally in surrounding habitat may be more useful than corridors.

Boecklen and Gotelli (1984) found only half the variation in species richness to be explained by the species-area model and suggested that its predictive power can be improved by incorporating other measures of habitat heterogeneity or resource availability. Zimmerman and Bierregaard (1986) concluded that 100 ha containing quality breeding habitat would preserve more tropical forest frog species than 500 ha containing little or low quality habitat. Schroeder et al. (1992) found that including habitat conditions (e.g., foliage height diversity [MacArthur and MacArthur 1961]) provided more accurate predictions of breeding bird richness than area alone. Several wetland evaluations have already been mentioned that modify the area criterion value based on other habitat quality values. Durham et al. (1988) used three landscape variables (watershed quality, interconnectedness, and interspersion), other than size, to measure habitat quality of bottomland forests in their community model.

Contiguity and adjacent land cover and use are prevalent in wetland evaluations but are justified or interpreted in many ways. Golet and Larson (1974) consider "natural" habitat to serve as a buffer and reduce human disturbances affecting wildlife (although benefits to waterfowl are mostly cited). Gosselink and Lee (1989) consider upland habitat for its "accessibility," reasoning that movement and dispersal of wildlife into bottomland forests and along natural corridors and the potential for increasing habitat of stenotopic, interior species are critical. They suggest that 100-m buffers of upland forest are necessary for extending effective areas for interior species and 50-m corridors along streams are necessary to facilitate wildlife movement between dispersed tracts of upland forest. Ammann and Stone (1991) consider a 150-m buffer of undisturbed wildlife habitat necessary to provide benefits for wildlife movement and reduce human interference. Limiting human disturbance near nesting islands (for wading birds and black duck), riparian forest nest sites (for bald eagle), and open-water feeding and roosting sites (for diving ducks) is becoming an important consideration as shoreline development and boating activities increase (Erwin et al. 1993). Adamus (1983) considers land cover and use in the watershed in relationship to import and use of nutrients in wetlands for food chain support and production. Leidy et al. (1992) criticize the exclusion of adjacent areas in evaluation of jurisdictional wetlands under the Clean Water Act because all habitats along the moisture gradient play a crucial role linking aquatic with upland habitat, and are central to the overall health of both ecosystems.

Secondary effects of new or disrupted ecological interactions are difficult to document, but the effect on community structure of predation and competition is often suggested by study findings (Terborgh and Winter 1980, Wilcove et al. 1986, Whitcomb et al. 1981). Studies conducted in the prairie pothole region suggest that duck nest success may increase with range expansion of coyotes due to interference competition with red foxes, even though both depredate duck nests (Sovada et al. 1995). The combined nest success improvement (from 17% for areas with only red fox to 32% for areas with only coyote) is especially interesting because most areas with coyote had success well above the 15-20% suggested threshold for population stability of several dabbling duck species in the region.

Rarity

Dony and Denholm (1985) consider rarity to be a measure of quality because rare species often indicate unusual ecological conditions, are more vulnerable to human pressure, and are prone to extinction (Terborgh and Winter 1980, Margules and Usher 1981). Demographically, the probability of extinction of a local population declines as its size increases (Goodman 1987, Pimm et al. 1988). From a population genetics viewpoint, low densities mean higher probabilities of depleting genetic variation and lower chances of long-term survival (Lande and Barrowclough 1987). Much environmental variation is not correlated on a regional scale, and species present at many sites should have better chances of survival than species with restricted distributions (Goodman 1987, Arita et al. 1990). However, rarity is only one of many factors (e.g., habitat fragmentation, colonization ability, territorial behavior, intraspecific interactions, migratory behavior [Terborgh and Winter 1980, Lande 1988, Pimm et al. 1988]) influencing extinction.

Rabinowitz et al. (1986) categorized flora of the British Isles based on three traits shared by all species—geographic range, habitat specificity, and local population size (Figure 3). Smith and Theberge (1986) summarized five types of rarity as: 1) species that are geographically widespread but are scarce wherever they occur (with either a patchy or continuous spatial distribution); 2) endemic species with restricted geographic ranges; 3) disjunct populations that are geographically separated from the main range of the species; 4) peripheral populations that are at the edge of the species' geographical range; and 5) declining species that were once more abundant and/or widespread but are now depleted.

Rarity introduces a problem of scale; species that are common in a study area may be uncommon nationally, or vice versa. Restricting rarity to nationally rare species can obscure distribution patterns necessary to understand a species' ecological requirements and assess its local importance to an ecosystem (Dony and Denholm 1985). Graber and Graber (1976) considered species with less than 500 breeding pairs and one species with an endemic population in the state as rare enough to be placed in a special category, similar to the red flags discussed previously, and not given points. They reasoned that state protection should be provided even though some of these species are more common elsewhere because first, the genetic breadth of a species is the sum of the genetic breadth of the populations, and second, habitat alterations occurring in one state are often happening in adjacent states. If each state allowed its population to slip away, the inevitable result would be extinction of the species.

Rarity is not limited to the species level. Taxonomic distinctness (i.e., monotypic genus, species, or subspecies) is a criterion for ranking the listing and recovery actions of federally endangered and threatened species (Fay and Thomas 1983). States may have their own criteria for designating rare populations or species at risk (Millsap et al. 1990).

Rarity can also be applied at the community level. The Nature Conservancy's "elements of diversity" approach (Jenkins 1978) classifies and ranks natural communities according to rarity at state and global scales. The community-level inventory serves as a "coarse filter" intended to protect 85-90% of species without inventorying them. Community inventories are supplemented by rare species inventories that serve as a "fine filter" for those species missed by the community-level filter (Jenkins 1985). However, preoccupation with climatic relics (that is, communities formed under past climatic conditions, such as many bogs, fens, and prairie remnants) has been criticized since they require labor-intensive successional management (e.g., burning, mowing, or herbicide treatment) simply to be maintained as "living museums" (Noss and Harris 1986).

There is strong interaction between seral stage and rarity because early seral stages are often preferred by generalist species whereas mature communities support more specialist species (Harris 1984). Bowles (1963) [cited in Harris 1984:65] observed twice as many rare species (ranked by frequency of observation) in old growth forests as in salvaged log blowdown areas in the Cascades, and twice as many common species in the salvaged areas as in the old growth forests. An old growth bottomland hardwood forest in South Carolina was compared to clear cut and selectively cut portions of the same area. More specialist species, particularly cavity nesters, achieved their highest densities in old growth, and those that achieved higher densities in disturbed bottomland forests were widespread throughout the region (Hamel 1989).

Rarity is a relative term and its evaluation is particularly dependent on the existence and synthesis of regional-level information (Smith and Theberge 1986). Rarity assessments are often expressed as the number of rare species or features in an area (Smith and Theberge 1986) necessitating lists of species and features considered rare at one or a series of geographic scales [e.g., local, regional, and national (Klopatek et al. 1981, Dony and Denholm 1985, Natural Heritage Data Center Network 1993)]. But how is rarity quantified? Rabinowitz et al. (1986) used no objective boundaries between categories but instead relied on a survey format, asking biologists to classify species based on a few selected examples. Arita et al. (1990) used median density and area of distribution as the basis for rarity of neotropical mammals. In Great Britain, a national grid of 10-km squares is used to record the distribution of species, and the number of grids a species is recorded in may be the basis for designating rarity (Margules and Usher 1981).

Element ranking procedures of state natural heritage programs, developed through collaboration between The Nature Conservancy and state natural heritage programs, have emerged as a national standard for designating rare and imperiled resources based on global and state rankings (Pearsall et al. 1986). Ranks reflect a species' abundance (number of individuals), distribution (number of site occurrences), as well as imperilment by its fragility or threats it faces (Natural Heritage Data Center Network 1993). Ranks range from 1 to 5, representing a status of critically imperiled to demonstrably secure. In general, species with a rank of G1-G3 (Table 4) are biologically qualified for listing as endangered or threatened, subject to additional questions about degrees of threat (Natural Heritage Data Center Network 1993).

Most wetland evaluations (Table 2) consider rarity in the context of state or federal endangered or threatened species. Some consider their presence to be a "red flag" indicator paramount to other criteria. Cable et al. (1989) assign higher values to less common and rare species. Ammann and Stone (1991) consider exemplary communities based on state natural heritage inventory. NCDEHNR (1995) also suggests using the state's natural heritage inventory for species and habitat types in evaluations. In addition, major wetland types commonly containing rare/endangered/threatened species and/or having unique habitats are noted. OMNREC (1984) emphasizes this criterion more than any other evaluation by incorporating the following measures: rarity of wetlands as a feature in the landscape, abundance of a particular type of wetland in a region (see also Representativeness below), rarity of animals and plants, and rarity of their habitat for breeding, feeding, or migration both at a provincial and regional level.

Productivity

Productivity is a measure of the rate at which communities of organisms bind energy into various kinds of organic material. It may measure gross values (the rate at which energy is consumed) or net values (gross production minus respiration, or the rate that biomass is being formed and lost). Or it may be specific to trophic level or system component (e.g., primary, secondary, tertiary, detritus production). Standing crop can be a conservative estimate of annual net primary production for herbaceous species or communities that die back or lose most of their growth each year (Kibby et al. 1980).

The rationale for using productivity as a criterion is that areas of high productivity are unusual and often provide the energetic basis for production over a larger area (Smith and Theberge 1986). Many examples of diffuse processes in a compound landscape having a concentrated effect within a component subsystem involve productivity and its translocation (Harris 1988). Large, mixed-species colonies of wading birds may translocate 400 metric tons of mineral ash per year (Harris 1988). Exploitation of horseshoe crab eggs during a two week stopover period by migrating shorebirds in Delaware Bay may translocate 150 metric tons of eggs as fat (Myers 1986). The limited spatial-temporal scale of such dramatic natural events suggests that they may be more easily assessed in other ways (see Importance to Wildlife below). Translocations may also be detrimental, e.g., nonpoint source (agricultural runoff) and point source (mostly industrial and sewage treatment plants) pollution effects on abundance of submerged aquatic vegetation, waterbirds, finfish, and shellfish in Chesapeake Bay (U.S. Environmental Protection Agency [USEPA] 1982).

Much information has been compiled on the productivity and standing crop of different wetlands, their species and components, and transport between other systems (see Kibby et al. 1980, Adamus 1983, and Greeson et al. 1979). Such studies and direct measurement of productivity are time-consuming and expensive. Smith and Theberge (1986) suggest that productivity is correlated with other criteria. The importance of an area to a species is often related to the degree to which the area contributes to the species growth and reproduction (see Importance to Wildlife below). Productivity has also been argued to be correlated with diversity both positively (Connell and Orias 1974) and negatively (Huston 1979). Oviatt et al. (1977) showed that field measurements of productivity exhibited as much variation within a marsh from place to place or time to time as between marshes, and the variation appeared unrelated to disturbance. Additionally, lack of correlation among productivity parameters makes the utility of their measurement questionable and requires value judgments in selecting a particular indicator (Oviatt et al. 1977).

Few evaluation methods for natural areas use productivity as a criterion (see Table 3). Those that do use it are primarily applicable to aquatic systems (Smith and Theberge 1986). Five of the wetland evaluation methods consider productivity in some form (Table 2), but no standard concept has emerged for what should be measured. OMNREC (1984) suggests that primary productivity is a good indicator of overall biological productivity. Growing season degree days, soil characteristics, wetland type, site location, and nutrient status (total dissolved solids) are used to derive a subjective productivity value for a site. Golet (1976) uses maximum wildlife production (primarily waterfowl) but only considers the food resource as reflected by water chemistry (i.e., total alkalinity and pH). He also points out that maximizing diversity and production for wildlife are reasonable goals, but they are not compatible.

Other methods emphasize food chain support as a major function of wetlands. Adamus and Stockwell (1983) define food chain support as "direct or indirect use of nutrients by animals inhabiting aquatic environments . . . primarily fish and aquatic invertebrates of commercial or sport value." From a wetland productivity standpoint, this measure reflects only the exported net community production (primarily detritus) used by a portion of the adjacent or downstream aquatic community—a difficult measure to assess scientifically and of dubious utility in wetland evaluations (Stuber and Sather 1984).

In summary, direct measurement of productivity appears to have little utility in wetland evaluations.

Importance to Wildlife

Three components of importance to wildlife are commonly considered: 1) the relative importance (i.e., suitability) of habitat to different stages of a species' life cycle (e.g., breeding, migration, etc.); 2) the relative importance of the site's population to some larger region or population; and 3) the relative importance (i.e., quality) of different species, say, from most endangered to most abundant and widespread (Smith and Theberge 1986). Areas or habitats that satisfy one or more of the above may be categorized by many adjectives such as suitable, quality, significant, sensitive, critical, or unique.

In general, no explicit method is used in selecting species of concern for evaluations. Evaluation may focus on game species (i.e., harvested waterfowl), other species identified in legislation (e.g., migratory birds, threatened, or endangered species), or consider traditional breeding sites, nesting colonies, staging areas, foraging areas, and molting areas. However, guidelines are available for species selection based on their characteristics that make them suitable for conservation in natural areas (Adamus and Clough 1978) or that make them suitable for indicating project impacts during environmental assessments (USFWS 1980, Roberts and O'Neil 1985).

NCDEHNR (1995) suggests measures of wildlife habitat quality (e.g., good cover and food) be subjectively rated. Municipality of Anchorage (1991) considers certain species to only occur in wetlands of a particular quality and uses presence of these species to specify wetlands deemed locally important for their breeding, feeding, spawning, or rearing. In contrast, the method's "waterfowl staging areas" criterion is based on numbers of species and overall abundance, while its "other migratory bird staging areas" criterion is based solely on species richness.

Two methods (Adamus 1983, Adamus et al. 1987) use simple descriptions of habitat to assess suitability for any of 14 groups of waterfowl during breeding, migration, and winter periods, or 120 species of wetland-dependent birds. High ratings imply that the site normally supports high levels of productivity for the species selected and may require confirmed sightings of the species during a field visit (Adamus et al. 1987).

Another method, the Habitat Evaluation Procedures (HEP) (USFWS 1980), relies on Habitat Suitability Index (HSI) models (Schamberger et al. 1982) to evaluate habitat quality for individual species of fish and wildlife. These models assess the ability of the habitat to provide the life requisites (e.g., food, water, cover) of the species. Selection of species is important because the selected species are often chosen to represent aspects of the larger system. HEP is widely used across North America and provides a consistent approach and repeatable methodology for various types of impact assessments, including activities in wetlands. The reliability of most HSI models is not known even though measures are clearly stated and based on extensive literature review. In addition, a complete HEP analysis can be fairly time consuming when compared to several of the wetland evaluation methods discussed here.

Special characteristics of single sites can render them uniquely able to support large numbers of migratory bird populations (Myers 1986). However, some wetland evaluations provide only guidelines for evaluating significance. Larson (1976) considers wetlands used by "great numbers of migrating waterfowl, shorebirds, marsh birds and wading birds" to have outstanding values but provides only a few sites in Massachusetts as examples. Bond et al. (1992) base significance on national and provincial inventories when available. OMNREC (1984) considers waterfowl staging areas, winter cover for wildlife, waterfowl production, nesting sites for colonial waterbirds, and migratory stopover areas (roughly in order of importance) at one or more geographic scales of significance, but provides few guidelines other than that appropriate agencies be contacted for information. None of these methods provides specific numerical criteria.

Szijj (1972) suggested that numerical criteria be considered to assess the international significance of wetlands for birds. Subsequently, 1% of a defined population, or a total count of at least 10,000 waterfowl or 20,000 waders (i.e., shorebirds, herons, and their allies), became widely accepted for evaluating wetlands as habitats for non-breeding waterfowl and colonial breeding birds (Fuller and Langslow 1986). The 1% criterion has also been applied to sites of national and regional significance (Fuller 1980, Lloyd

1984), although populations with fewer than 100 pairs might be reconsidered under the rarity criterion.

A number of problems arise when applying the 1% criterion. Accurate population estimates at the scales of interest must be available. Such data may only be readily available for nesting colonial birds (Osborn and Custer 1976). The 1% criterion is not suited for situations where habitat is extensive or birds tend to be dispersed, situations where density of individuals may be a more useful criterion (Fuller 1980). However, Fuller (1980) established criteria based on opinions of ornithological experts when no adequate national population data exist. Finally, peak population counts of migrant or wintering birds do not give a true reflection of the numbers of birds using a site if there is a turnover of individuals in the population (Fuller and Langslow 1986).

Graber and Graber (1976) used a faunal index to score breeding bird species based on their abundance statewide. Introduced species were specifically excluded. Scores roughly double as statewide breeding bird populations geometrically declined below 50,000 birds. Species with less than 500 breeding birds and one species with an endemic population in the state were considered too rare to assign points. In addition, a high degree of habitat specialization doubled the points assigned a species. This approach was adopted by Cable et al. (1989) who adjusted scores to a more restricted range of abundance (from 5,000 downward to 50 individuals) and only considered wetland-dependent breeding bird species. The index was further adjusted based on an optimal species-area relationship (see Area above).

Millsap et al. (1990) used a hierarchical approach, based on biological, management, and other considerations, to rank species for setting conservation priorities in Florida.

Biological variables included overall population size and trend, distribution and its trend, population concentrations, reproductive potential, and ecological specialization. One or more combinations of these scores can be used as an indicator of species quality and applied to the occurrence or abundance of species at an evaluation site.

Finally, critical habitat is a designation related to federally listed threatened and endangered species. It may be the "entire species habitat or any portion thereof, if, and only if, any constituent element is necessary to the normal needs or survival of that species" (Basinger 1980). Only 12% of federally listed threatened and endangered species have critical habitat designations (50 CFR § 17.11-12), but any evaluation that considers threatened or endangered species must also be aware of these critical habitat designations.

Representativeness

Representativeness has been described from two rather different viewpoints. The more common view, referred to as inclusive representation (Smith and Theberge 1986), approaches selection of natural areas as a means to represent the full range of natural features in a system of reserves throughout some larger geographic region. It is global in context and involves specification of the region of interest, a biological classification of the region, and a regional inventory (e.g., vegetation mapping) as preliminary steps to a site's evaluation (Austin and Margules 1986).

In contrast, typicalness, the other view of representativeness, is not a global concept and can be assessed for an individual site. It may reflect the degree to which a site's habitats, communities, and species are commonplace, or how representative the site's composition is of some biological description, such as an ecosystem (Usher 1986) or community type (Noss 1987). Duever and Noss (1990) refer to this as completeness, and define it as a site with representative communities with natural, diverse habitats and a full complement of species. Typicalness may be considered the opposite of uniqueness (Smith and Theberge 1986). This creates some confusion in the use of rarity and typicalness as criteria. A site with many rare species can not be typical, and a typical site can not have more than an average number of rare species (Usher 1986).

On the other hand, Austin and Margules (1986) suggest that the less natural a landscape the more prominence given to diversity and rarity, while the more natural a landscape the greater the emphasis on representativeness in assessing conservation values. Similarly, three degrees of habitat representation in protected areas were noted by Moore (1987), relating representativeness to threat and human interference. The situations he distinguished were: 1) the habitat is extensive, and the protected areas are only examples of it; 2) the habitat area is reduced, so the protected proportion increases even though the protected area has not changed; and 3) the protected area is increased to protect what is left of the habitat, most of which is now parks or reserves. Thus, an inclusive view may anticipate threat of human interference rather than simply reacting to it. In any event, representativeness, much as naturalness, is often based on a time prior to massive urban/industrial/agricultural change (van der Ploeg 1986).

Margules and Usher (1981:100) claimed that the inclusive viewpoint subsumes typicalness:

> Areas selected to be representative would necessarily include typical and common species, but they could also include rare species since the objective is

to represent the full range of biota.

By either definition, the issue of representativeness is essentially one of classification (i.e., determining the range of natural features in a geographic region and which of those features are exhibited at a particular site). A description of representative categories is often a preliminary step in evaluation (Austin and Margules 1986, Crumpacker et al. 1988), and classification may serve as the primary basis for the evaluation (Golet 1976). Unfortunately, in order to assess the extent to which an area represents the flora and fauna in a region, the analysis, or at least a classification, must be extended to include that whole region. A context must be provided for the evaluation of a site and the region must be defined (Austin and Margules 1986). Thus, representativeness is central to the issues of wetland delineation (Leidy et al. 1992) and regionalization of evaluations (Stuber and Sather 1984) where classification constrains those sites that may be compared. (See USACOE 1988 as an example of regionalization.)

Representativeness as a goal in conservation is viewed as necessary to maintain biodiversity in functional ecosystems (International Union for Conservation of Nature and Natural Resources 1978). The rationale for using representativeness is that it would be naive to think we know which component of habitat is more significant than another, or which elements of the habitat will be of great future value to man (Smith and Theberge 1986). This view is embodied in The Nature Conservancy's use of communities as elements of biodiversity. An ideal goal for a state heritage program might be to protect the best examples of each major community type in each physiographic region in the state (Anderson 1982). Representatives of all native communities then serve as a coarse filter to protect and capture perhaps 90% of all species without having to inventory them individually (Noss 1987).

An ecosystem is usually broadly defined as a biotic community and its environment (Odum 1971). It can be described at various hierarchical levels from the biosphere down to small, statistically recognizable plant associations (Daubenmire 1968) or aquatic dominance types (Cowardin et al. 1979). Crumpacker et al. (1988) used Kuchler's Potential Natural Vegetation to examine the representativeness of Federal lands for conservation on a national scale. They concluded that 33 of 135 classes were poorly represented on federal lands, nine of which had no representation.

More recently, Gap Analysis uses vegetation maps as surrogates for ecosystems in conservation evaluation and seeks to identify vegetation types and species that are not represented in the network of biodiversity management areas (Scott et al. 1993). Its overall cover classification provides hierarchical groupings for natural terrestrial cover (Jennings 1993), natural aquatic cover (Cowardin et al. 1979), and cultural, or developed, cover (Anderson et al. 1976). Natural terrestrial cover types follow a United Nations Educational, Scientific and Cultural Organization (UNESCO) format as modified for the United States by Driscoll et al. (1984) using four physiognomic and two lower floristic levels (Figure 4). The two lower floristic levels—cover type and community type—represent actual or existing vegetation, rather than potential vegetation of Kuchler or climax seral categories of the UNESCO plant series and associations (Jennings 1993).

Gap Analysis is typically being conducted on a statewide basis and uses a 100-ha minimum mapping unit (e.g., the size of the smallest polygon mapped), in uplands, and a 40-ha minimum mapping unit in wetlands and riparian areas. This level of detail is not adequate to understand the representativeness of wetlands, which must be inferred from association with other habitat types or based on comparable scale hydrography representing streams and lakes (Scott et al. 1993). But even these methods do not adequately represent species (e.g., amphibians, mallards [*Anas platyrynchos*], song sparrows [*Melospiza melodia*], water shrews [*Sorex palustris*], and muskrats [*Ondatra zibethicus*]) that use smaller wetlands.

National Wetlands Inventory (NWI) maps (1:24000 scale) can be used as a basis for assessing wetland representativeness (USFWS 1990a). Unfortunately, statewide coverage of these maps is not always available, and their manipulation for statewide evaluation is unwieldy (Scott et al. 1993). Bara (1994) describes use of Thematic Mapper satellite imagery in five additional federal environmental monitoring programs—Environmental Monitoring and Assessment Program (EMAP) of USEPA, Gap Analysis Program of the National Biological Service, National Water Quality Assessment Program of the U.S. Geological Survey (USGS), CoastWatch Change Analysis Program of National Ocean and Atmospheric Administration, North American Landscape Characterization Project of USEPA and USGS. These may also serve as good sources for evaluating representativeness or monitoring change but at less resolution than provided by NWI data. Nevertheless, these programs can be useful for assessing landscape level concerns about included wetlands.

The USFWS elaborates the "no net loss of wetlands" policy as meaning that wetland losses must be offset by wetland gains in terms of acreage and, to the extent possible, ecosystem function (USFWS 1990b). Thus, it is critical to know the representativeness of wetlands that are lost regionally (see also Fragility and Threat below). Wetland status and trends reports (Frayer et al. 1983, Dahl and Johnson 1991) may be used to assess regional wetland types most threatened with losing their representativeness. Since 1780, 53% of the wetlands in the conterminous U.S. are estimated to have been lost (Dahl 1990). Dahl and Johnson (1991) noted that 98% of the decline in wetlands from the mid-1970s to mid-1980s occurred in freshwater wetlands. However, this didn't express the additional 800,000-acre loss of palustrine wetlands offset by a gain in non-vegetated wetlands, primarily ponds, not previously classified. Graber and Graber (1976) defined an "acreage factor" as the reciprocal of the proportion of a habitat in the region [state] plus the percent change of habitat type over a decade. This index could be calculated from NWI status and trends data (Dahl and Johnson 1991). However, Frayer et al. (1983) recommended that their state estimates not be used for making decisions, since many estimates, especially for rarer wetland types, are subject to a high degree of sampling error for smaller regions.

Most wetland evaluation methods incorporate representation in a reactive context by emphasizing scarcity (see Rarity above) or loss of particular wetland classes within a watershed, region, or state. For example, Municipality of Anchorage (1991) uses a scarcity value (acreage of wetland type on the site relative to its total in the catchment basin); Larson (1976) considers scarcity of wetland types within physiographic regions; and OMNREC (1984) considers overall scarcity of wetlands and wetland types within physiographic regions. Bond et al. (1992) consider development pressure on wetland types within 20 wetland regions in Canada. Adamus et al. (1987) use a value similar to Municipality of Anchorage, but compare it to the statewide loss rate for the wetland type. Gosselink and Lee (1989) use two measures that could be considered as components of representativeness—fraction of bottomland hardwoods remaining in a watershed, and their patch size distribution (Table 5).

Naturalness and Ecological Integrity

Naturalness connotes an abstract state that may seem difficult to quantify. Part of the difficulty is that naturalness represents a dynamic state of a functioning ecosystem. For example, it implies natural regimes of disturbance such as windthrow, fire, and flooding (Smith and Theberge 1986). In large tracts, natural disturbance creates a shifting mosaic steady state (Noss 1983). Natural disturbance of riverine channels maintains portions of the floodplain in early seral vegetation and enhances diversity in plant communities, structural composition, and the resulting wildlife community (Cooper et al. 1990). Where processes are allowed to continue, the diversity of communities will persist. However, normally natural disturbances can pose a threat in a fragmented landscape where the shifting mosaic has virtually nowhere to shift (Noss 1983, see Size and Landscape Pattern above).

Margules and Usher (1981) suggest that naturalness implies freedom from human influence, but in practice the criterion represents the "degree of influence" since few, if any, sites are free from human activities. They further suggest that humans can be an integral part of natural ecosystems provided that, if they are present, they are totally dependent upon and limited by their local environment (Margules and Usher 1981). Consequently, natural conditions are frequently defined as the state existing at the time of European settlement (Bonnicksen and Stone 1985). Nevertheless, Anderson (1991) considers over exploitation by aboriginal man an unnatural impact, and Simenstad et al. (1978) documented structural changes in marine ecosystems as a consequence of over-exploitation of sea otters by the Aleut.

Human activities, and therefore naturalness by the definition above, may be assessed directly or indirectly in many ways. The degree of human influence may be explored by suggesting changes that would occur if humans, or their influences, were removed (Anderson 1991). The cultural energy necessary to maintain a functioning ecosystem may be quantified and used as an indicator of human disturbance because energy subsidies often mask degradation of ecosystems (Rapport 1989). Management practices can be compared to natural disturbances to assess whether systems are pre-adapted to handle imposed stresses (Westman 1985). For example, fire management is more natural than selective logging which is more natural than clear cutting; and agricultural systems requiring large amounts of herbicides, fertilizers, and fuels for farm machinery can be considered artificial ecosystems (Anderson 1991).

Human disturbances may also be classified to indicate their potential severity. For example, some management practices (e.g., burning, mowing) and land uses (e.g., agriculture, forestry) may be more compatible than others. In turn, these are always less severe than total replacement (e.g., mining, damming) (see also Landscape Patterns above). NCDEHNR (1995) uses measures of disturbance—including the degree to which surface drainage is regulated, as well as the severity and duration of human and natural disturbances—to assess a wetland's status rather than value *per se*. Gosselink et al. (1990) used a multi-factor synthesis (area disturbed and the intensity and permanence of the disturbance) to assess human activities and their consequence to ecological attributes of bottomland hardwoods. They concluded that mining had the most severe effect but that water resource development activities (channelization, impoundment, and levee construction) are as significant because of their pervasiveness throughout the watershed. Durham et al. (1988) use a watershed quality index that assesses the amount of disturbance in the watershed resulting from urban, industrial, and agricultural activities.

In human dominated landscapes, naturalness has often been correlated with the occurrence of exotic, or introduced, species (Smith and Theberge 1986). Anderson (1991) recommends an index of the proportion of native species that remains in an area. This index is not unlike one used by Newmark (1987) who inventoried the mammalian species in 14 individual or complexes of national parks to rank them on the basis of the proportional loss of species since European settlement. Alternatively, an index representing the proportion of native species in the total present day assemblage [i.e., (no. of native species)/(no. of native + exotic species)] could be used if presettlement conditions were in doubt (Anderson 1991). Rapport (1989) suggested this index as one measure of ecosystem health and integrity. Presence, however, does not necessarily represent the "natural ranges of abundance," and perhaps species that are below a minimum viable population estimate (Shaffer 1981) or are listed as endangered should also be counted in the proportional loss (Anderson 1991).

More emphasis should be placed on maintaining physical habitat conditions and ecological processes rather than looking for single causes (Noss 1990). Rapport et al. (1981) suggest that the diagnosis and rehabilitation of an ecosystem’s health is similar to human medical practices because the central question concerns which stresses the system can cope with by its own regulatory mechanisms and which lead to breakdown of system processes. The analogy to human health has also been extended to landscape health where homeostasis is characterized by 1) the self-maintaining balance of biotic communities to recover from stress and disturbances, and 2) the dynamic equilibrium of streams between the inflow and discharge of water and sediments (Ferguson 1994). Therefore, diagnosis, prognosis, and treatment of ecological health rest on: 1) identification of critical characteristics that differentiate healthy ecosystems from sick ones (e.g., maintaining efficiency in energy transfer and nutrient cycling, and maintaining a diverse species assemblage in which the longer-lived and larger life-forms are dominant in the mature phase of ecosystem development), 2) measurement of the counteractive capacity to handle stress loading (i.e., the system's ability to bounce back, or recover from perturbations), and 3) the risk or threat posed by exposure to certain sources of anthropogenic stress (Rapport 1989, see also Ecological Fragility, Replaceability, and Threat below). Such qualitative assessments of changes in magnitude and direction of critical system functions and components may benefit from systemic indicators of ecosystem integrity (see Odum 1985, Rapport 1989).

Ecological integrity is important in protecting biodiversity (Noss 1990, Karr 1990) and has been related to the minimum critical size of ecosystems (Lovejoy and Orens 1981). Anderson (1991) suggests that natural systems possess ecological integrity and are intact. For example, Karr and Dudley (1981) define biological integrity as the capability of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition and functional organization comparable to that of natural habitat of the region. The concept requires an integrative approach since the limits to biological integrity vary with type and magnitude of human impacts in both spatial and temporal scales (Karr 1990). An index of biological integrity (Karr 1981, Karr et al. 1986) for streams assesses five indicators—water quality, flow regime, energy source, biotic interactions, and habitat structure. Twelve metrics are used to evaluate a single sample from a stream reach and determine the extent to which the resident biotic community diverges from that expected of an undisturbed site in the same geographic area and of the same stream size. The index can be used to assess local conditions or entire watersheds (Steedman 1988).

A conceptually similar approach is proposed by Gosselink and Lee (1989) for bottomland hardwood landscapes. They describe cumulative impacts derived from multiple activities in the landscape where the major long-term effects are: 1) forest clearing, reducing the total area of bottomland hardwoods and fragmenting remains into smaller tracts, and 2) hydrologic modification for flood control, navigation, and electric power, resulting from construction of numerous dams, thousands of kilometers of levees, and dredged streams. They propose using nine indicators of ecosystem integrity (Table 5) and long-term data sets to assess or monitor cumulative impacts in areas the size of entire watersheds (c. 1,000,000 ha). The assessment relies on many of the principles of island biogeography already discussed plus the water-dependent bottomland hardwood forest functions related to hydrology, sediment transport, and water quality. Definitions of integrity typically are based on an implicit understanding of naturalness (Anderson 1991). For example, Gosselink and Lee (1989) provide rationale that flora and fauna decline without the flooding regime of a natural stream.

Bond et al. (1992) use wetland viability criteria (including cumulative impacts, ecosystem functional status, and potential for restoration) for assessing development actions. However, the rationale is, if the wetland is currently degraded and can not be reasonably rehabilitated, then projects should only be of concern if the wetland is regionally rare. In other words, it is a permissive, not restrictive, criterion. Other methods (Adamus 1983, OMNREC 1984, USACOE 1988) consider hydrologic criteria (e.g., flow stabilization, sedimentation and nutrient trapping, water quality improvements) in the context of direct social benefits, rather than indirect benefits accrued from naturally functioning, healthy, non-degraded ecosystems with intact community structures. Consequently, these scores are highest (i.e., best) when the impact to healthy, intact ecosystems is greatest.

The description and use of naturalness and integrity are ecosystem dependent. The value of naturalness is that undisturbed sites provide the best source of baseline information to compare with other modified areas (Jenkins and Bedford 1973). On the other hand, ecological integrity, although just as abstract as naturalness, avoids the more philosophical justification of its use based on preservation of wilderness (Smith and Theberge 1986). One way to distinguish the two conceptually might be to consider naturalness as representative of a system's state while integrity focuses on a system's processes, although the literature implicates both structure and function when either is discussed (Karr 1990, Anderson 1991). It is important to note that their use in wetland evaluation may be limited since evaluations are mostly a snapshot of a site and the criteria frequently depend on long-term, geographically extensive data sets for their description (Gosselink and Lee 1989). But, while evaluation primarily focuses on structural measures (diversity, landscape pattern, rarity, representativeness), the functional characteristics will prevail in the long term. What good is it to preserve diverse productive bottomland hardwoods if the hydrological regime precludes regeneration? Nevertheless, evaluation used to monitor the status of ecosystems can be helpful to detect long-term unanticipated responses to functional changes in the system.

Ecological Fragility, Replaceability, and Threat

Ecological fragility describes communities with an intrinsic sensitivity to change (Margules and Usher 1981) and may be viewed as a gradient whose other end is community stability (Smith and Theberge 1986). Another criterion, replaceability, is closely related and can be defined as the ability of an ecosystem or population to return to its original state after a specific disturbance (van der Ploeg 1986). However, these criteria are conceptually complex since there are a variety of dynamic properties of stable-state ecosystems as they respond to disturbance. The ability of a system to resist change may be described by its persistence [e.g., the duration a variable remains unchanged during a perturbation (Pimm 1984)], and its inertia [e.g., how much a variable changes following a perturbation (Westman 1978)]. The ability of a system to recover from a perturbation, or its resilience, may be described by its: 1) elasticity, the pace of recovery, 2) amplitude, the magnitude of displacement that can be tolerated, 3) hysteresis, the differences in the paths of alteration and recovery, and 4) malleability, the degree to which the new stable state differs from the original stable state (Westman 1978).

The emphasis of fragility on disturbance relate this criterion to ecological integrity, while the response (i.e., recovery) is related to replaceability. Fragility and replaceability apply easily to climax communities since they are not expected to change unless there is some change in the physical environment or land use (Margules and Usher 1981). Climax communities are generally composed of longer-lived life forms than earlier seral stages (Whittaker 1975), and thus are usually more persistent but less resilient (Pimm 1984). Using similar rationale, Suffling (1980) estimated elasticity based on the ecological age or history of communities at a site. Graber and Graber (1976) defined this as replacement cost, or the time required to reestablish a particular community, calculated as the age of the habitat in years plus a successional lead-in period. However, they concluded that man-made marshes should be dated historically, while natural marshes should be given higher replacement values than reflected in the individual marsh plants because of their long developmental history dating back to glacial times and manifested to some extent in underwater soil development. OMNREC (1984) uses ecological age (defined as a community's replacement time), arguing that destruction of a bog community would leave those species dependent on it without habitat for hundreds of years while a marsh could be reestablished and provide habitat within years or decades. In contrast, the system property of inertia appears to be difficult to quantify, being dependent upon the particular variable assessed and the system having not undergone structural change (Harwell et al. 1977).

Considerable information exists about restoring and creating wetlands (Schneller-McDonald et al. 1990). Opportunities for restoring wetlands vary with type of wetland and setting. Restoring vegetation in coastal and estuarine wetlands is often the immediate objective (Zedler and Weller 1990). Efforts to restore riverine wetlands are complicated by hydrologic and sediment regimes, making it difficult to restore wetlands without removal of structures and channels responsible for the original losses (National Research Council 1992). Large freshwater wetlands altered for agriculture but still poorly drained with largely intact hydric soils and hydrology (e.g., floodplains or deltas, pocosins of the southeast, everglades, alluvium in front mountain ranges) constitute the largest area of potential restoration (National Research Council 1992). Most wetland scientists agree that restored or constructed wetlands re-create functional ecosystems and maintain regional biodiversity (Zedler and Weller 1990); however, replacement is contingent upon an absolute and irrefutable requirement—that the living components of the community have not been removed to extinction (Graber and Graber 1976).

By far the most prevalent means of assessing fragility is in relation to a particular type of disturbance and the features thought to be most affected (Smith and Theberge

1986). Cairns and Dickson (1980) used the concepts of resistance and resilience to develop a method of qualitatively estimating "ecosystem vulnerability" in streams subjected to environmentally catastrophic events (e.g., spills of hazardous wastes). On the other hand, it is this disturbance dependence, where the same system may exhibit different responses to different disturbances, that makes the concepts of resistance and resilience so difficult to interpret (Gigon 1983). Thus, the concepts are more relevant to impact and risk assessment than wetland evaluation. Furthermore, the fact that many systems are exposed to multiple impacts should favor attention focused on biological integrity (see Naturalness above).

Threat of human interference has frequently been used in early evaluations of natural areas, but not explicitly in any wetland evaluations (see Tables 2 and 3). In addition, fragility in evaluation of natural areas is most often related to human induced perturbations (Smith and Theberge 1986). Duever and Noss (1990) refer to this criterion as vulnerability, defining it as the likelihood of events that might degrade or destroy the site occurring within the next few years.

Margules and Usher (1981) claim that the criterion is not based on any ecological principles. However, they note its assumed importance when species or communities are rare or lack resilience, and state (p 100):

> Species which are rare are often considered to be under the threat of human interference simply because they are rare. Species occurring in only one or two locations, even in some numbers, are again considered at risk because of human interference since there are few, if any, sources of immigration or recolonization.

In such instances, the criterion may actually be redundant with rarity and count the same factor twice.

Smith and Therberge (1986) consider threat as a planning and management criterion rather than a biotic one. The aspects of threat most often considered are severity and imminence (Smith and Therberge 1986). This context is quite similar to criteria used for listing species as federally threatened or endangered where both magnitude and immediacy of threat are considered (Fay and Thomas 1983). The degree of threat is also used to rank recovery actions for listed species (Fay and Thomas 1983). Therefore, this criterion is implied in all wetland evaluations that consider federally listed threatened or endangered species.

Mace and Lande (1991) argue that threat of extinction is a scientific criterion that may be based on a species' probability of extinction within a given period of time. Extinction may be deterministic or stochastic in nature. Deterministic extinction occurs when something essential is removed (e.g., space, shelter, or food), or when something lethal is introduced (e.g., feral cats or selenium). Stochastic extinction results from normal, random changes or environmental perturbations. Usually such perturbations do not destroy a population, but if the population is small, the interval of perturbation is short, or the perturbation is unusually catastrophic, the species' vulnerability to extinction is great (Gilpin and Soule 1986). Small, in this context, has been estimated in the range of 50 individuals in the short term, or 500 in the long term. These estimates may be too optimistic based on either demographic (Shaffer 1981) or genetic considerations (Franklin 1980).

Population viability analysis identifies and quantifies risk factors related to population dynamics, population characteristics, and environmental effects (see Table 6 and Gilpin and Soule 1986). Mace and Lande (1991) propose using such analyses and models that consider all the various extinction factors and their interactions, and define three categories for risk of extinction (not including extinction) as: 1) critical—50% probability of extinction within 5 years or 2 generations, whichever is longer, 2) endangered—20% probability of extinction within 20 years or 10 generations, whichever is longer, and 3) vulnerable—10% probability of extinction within 100 years. They also provide more qualitative criteria—including effective and actual population size, degree of population fragmentation, and population trends and fluctuations—because the data upon which to base calculation of extinction probabilities, in fact, rarely exist.

Masters (1991) suggests that the procedures of state natural heritage programs (see rarity above) are an alternative to the above approach. Elements of natural biodiversity, primarily species, intraspecific taxa, and natural communities, are assigned conservation priority ranks. A 1 to 5 rank (Natural Heritage Data Center Network 1993) is used for all elements at the global (rangewide), national, and state or provincial level based on objective information about: 1) population size, 2) range, 3) habitat specificity, 4) population and distribution trends, 5) threats to survival, and 6) fragility. The combined global, national, and regional ranks then give an instant summary of an element's known or probable threat of extinction or extirpation in a particular jurisdiction. Furthermore, in a jurisdiction where many highly ranked elements occur, the fact pattern used for an element's rank may be reviewed to distinguish and prioritize similarly ranked elements. Natural heritage procedures employ criteria similar to Mace and Lande (1991) but do so without specifying numerical thresholds that must be satisfied or combinations of factors that must be considered. Thus, procedures are flexible enough to be applicable across the entire spectrum of species and communities. Natural heritage procedures have also stood the test of time, having remained unchanged in its basic tenets for more than a decade (Masters 1991). Table 4 defines the three ranks biologically qualified for listing as endangered or threatened, subject to additional questions about degree of threat. Natural heritage procedures have been used to highlight the degraded condition of aquatic ecosystems in the U.S., where 45% of aquatic species are ranked G1-3 compared to only 12% of terrestrial vertebrates in the same categories (Natural Heritage Data Center Network 1993).

Contrary to Usher's (1986) belief that decline in the use of the "threat of human interference" criterion means that conservation is becoming more proactive, it would appear that the particular types of threat are being incorporated into other criteria, such as rarity, size, landscape pattern, and ecological integrity.

Discussion

Margules and Usher (1981) recognized that ecological criteria could be used in a site visit or regional inventory, or inferred from a case study (Table 7). This classification should be helpful to identify the context and relevance of criteria when setting objectives for ecosystem management. Given the breadth of criteria interpretation in current evaluation studies, not all criteria precisely fit this framework. Importance to wildlife may be evaluated using habitat models for individual species at a site, or may be evaluated relative to the site’s contribution to the regional or global population. Similarly, threat of human interference might relate to all situations, depending on whether potential consequences have already been studied elsewhere and whether the threat is viewed as a local phenomenon or regional activity.

Most regulatory agencies evaluate individual wetland sites as permits are requested. Information used to make recommendations is often obtained from brief field visits, a smattering of surveys, and recollections of office staff (Banner 1991). Thus, recommendations are typically subjective and can not account for cumulative impacts (Gosselink and Lee 1989). Golet (1979) presented several reasons for wetland evaluation and concluded that the most effective use will be in planning. He suggested that the relative importance of criteria be assessed and only the most important applied to quickly inventory all wetlands within a large geographic region. Conservation evaluations generally attempt to do this by examining entire landscapes, looking for sites with the highest conservation value. Conservation of natural areas is critical in both natural and human-dominated landscapes. While opportunities for protection and preservation are greatest in natural landscapes, regulation and restoration are the main strategies for maintaining healthy and self-sustaining ecosystems in human-dominated landscapes (Smith and Theberge 1986). Thus, regulation and restoration are the main strategies for conserving wetlands, and regionwide information will be necessary, but not sufficient, for success.

USFWS field offices are well aware of these issues, and alternative approaches are being sought to both rapidly screen permits and map sensitive resource areas (Banner 1991, 1992). Advanced identification provisions of the guidelines implementing Section 404 (40 CFR § 230.80) allow USEPA and the USACOE to identify critical wetland areas as generally unsuitable for use as a disposal site before any permit is requested. Such advanced identification studies can lead to a regional regulatory plan for achieving no net loss (Raines et al. 1990) and regional inventory maps to expedite permit review (USEPA 1990). Methods to screen permits only quantify values to fish and wildlife placed at risk. Permit applications that can not be quickly classified for acceptance or denial must be subjected to an impact assessment technique where mitigation actions may be determined. Screening methods must react to the scale and needs for evaluating a particular site. But by placing the site in the context of a larger landscape or watershed in which it exists, screening methods can be used proactively to set priorities and monitor trends. They may also be used to allocate time for protecting and mitigating impacts to the most valuable areas.

Setting objectives is a prerequisite of ecosystem management, especially as we consider larger and more complex natural systems (Schroeder and Keller 1990). Criteria for screening permit applications should also reflect regional conservation objectives, and it is not reasonable to expect criteria to be applied uniformly nationwide. Wetlands are diverse systems subjected to different demands and responding to alterations in different ways. Current methods demonstrate that it is feasible to assess wetland values at a site and compare similar sites in a region, but it is difficult to compare different types of wetlands or wetlands in different regions. Thus, evaluation methods have undergone modification to adapt them to a diversity of wetland systems and geographic regions (OMNREC and Environment Canada 1984, USACOE 1988, Gersib et al. 1989, NCDEHNR 1995).

To some extent, all ecological criteria are related to maintaining biodiversity. Pragmatic application of ecological criteria in screening will most often be determined by data availability. NWI data can be summarized, much as Golet (1979) proposed for regional planning, based on wetland class richness, dominant wetland class, and size.

Specifically, a geographic information system could provide NWI data summaries by watershed for the area of each wetland class present, the polygon size frequency of each class, and the degree of isolation. But use of NWI data has generally been limited to individually large projects or environmental assessment of estuarine development areas (Stayner et al. 1990). Geographic information systems have been used to evaluate habitat suitability for priority species over large regions in Florida, and their use has also been proposed for designating zones based on USFWS mitigation Resource Categories (Banner 1992, see Table 8). Regional maps of rarity and diversity may also be developed from existing databases (e.g., state natural heritage data and Gap Analysis). NWI maps can be used to address regional representativeness of the most valued wetlands (USEPA 1990). The goal of no net loss of wetlands will require monitoring to detect wetland trends and to guide timely adjustments in protection and management were wetlands are threatened. Wetland representativeness as well as the history and location of permit records (e.g., number, size, and proximity of permits previously approved) are critical elements for measuring achievement of this goal. Digital data is rarely managed by USFWS field offices, and must either be acquired or developed from information supplied by other agencies (e.g., state fish and game agencies). Gosselink and Lee (1989) emphasize the need for cooperative planning, common objectives among involved agencies, and improved institutional memory to maintain and update long-term data sets. Such actions are amenable to advanced identification of disposal areas (Raines et al. 1990). Similar challenges exist for the development and use of databases to support permit screening activities using ecological criteria.

Recommendations

Wetland evaluations for review of permit applications should be considered in a regional context to protect biodiversity and other regional ecosystem goals. This requires: 1) some type of long-term regional inventory and monitoring program, 2) improved record keeping, and 3) improved agency collaboration and data sharing. Much existing inventory data, particularly NWI, are underutilized due to their restricted accessibility (i.e., existence in a non-digital form). Other long-term monitoring programs (e.g., Gap Analysis, EMAP, CoastWatch) may also satisfy some regional inventory needs. With these thoughts in mind, the following recommendations are offered for selection of ecological criteria for use in wetland evaluations.

- Ecological criteria and their measurement must be flexible enough to address regional conservation priorities and use available information.

- Ecological criteria and how they are measured are more important than methods of scoring and comparing sites. A single overall index is less desirable than maintaining scores for individual criteria or wetland functions.

- Landscape patterns are important, being recognized to some degree in nearly all evaluation methods. Effective interpretation of patterns requires use of spatial analytical tools found in geographic information systems.

- Geographic information systems can be efficient tools for recordkeeping and long-term monitoring activities. However, improved agency collaboration and data sharing are necessary to realize such benefits.

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Table 1. Summary of 14 wetland evaluation methods reviewed for ecological criteria.

Adamus 1983 - Evaluation uses 11 functional values including wildlife habitat based on use by waterfowl and wetland dependent birds. Uses interpretive keys and multiple levels to assign probability ratings for social significance, and a wetland's effectiveness and opportunity to perform functions. No numerical value or overall probability rating is assigned. Intended for nationwide use to compare similar wetland types within a region.

Adamus et al 1987 - Revision to Adamus 1983. Wildlife diversity and abundance value based on interpretive keys for 14 waterfowl groups and 120 species of wetland dependent birds.

Ammann and Stone 1991 - Evaluates 14 functional values including ecological integrity and wetland wildlife habitat. Used on a local scale for wetland comparisons. Quantitative measures and wetland characteristics are subjectively scored and averaged to provide functional value index. Scores for each functional value are used separately.

Bond et al 1992 - Eliminative process at three levels of consideration. Structured to provide functional values (including ecosystem health and production) at national, provincial, regional, and local scales relative to development project characteristics. Initial level considers subjective measures for wetland viability, significance of habitat, and rarity with only broad guidelines provided.

Cable et al 1989 - Evaluates avian community of wetlands based on index combining diversity, rarity and size. Intended for regional comparisons among similar wetland types. Includes "red flag" elements to override significance of index.

Durham et al 1988 - Community model for bottomland hardwoods to assess their habitat functional value using criteria of diversity, size, landscape pattern, and naturalness. Requires significant data for 9 plot variables and 5 tract variables.

Golet 1976 - Uses 10 measures to evaluate maximum wildlife productivity and diversity criteria based on a classification developed by Golet and Larson (1974). Subjective scores for measures are weighted in overall cumulative score.

Gosselink and Lee 1989 - Proposes eight measures to assess ecosystem health and biological integrity of bottomland hardwoods throughout an entire watershed. Based on cumulative impact assessment of many human activities, no single one particularly large or damaging, but in sum total are both significant and dramatic.

Hollands and McGee 1986 - Uses a classification scheme to rate ten functional values including biological criteria attributed to Golet (see Golet 1976). Individual site score is cumulative across values and may be compared to mean model value, with other sites, or average of sites in a region.

Larson 1976 - Eliminative process at three levels using a classification scheme to screen and rate three functional values. Level 1 includes "red flag" criteria for rarity, abundance, and size that merit wetland preservation. (see Golet 1976 for Level 2 description)

Municipality of Anchorage 1991 - Similar to Ontario's method with biological values separated into habitat and species components. Many criteria may be attributed to Golet 1976. It also includes "red flag" elements for rare or significant species occurrences that elevate the importance of a site. Subjective scores are cumulative for each component; no overall score is used.

Table 1 (continued). Summary of 14 wetland evaluation methods reviewed for ecological criteria.

North Carolina Department of Environment, Health, and Natural Resources 1995 - Combination of unscored concerns and three functional values including ecological and landscape components. Criteria values are measured subjectively with few guidelines; scores are summed for overall wetland score. A simple method intended for statewide use.

Ontario Ministry of Natural Resources and Environment Canada 1984 - Complex subjective scoring of many factors which are summed into biological, social, hydrological and special feature indices. Used for regional comparisons of wetlands in Ontario. Biological criteria may be attributed to Golet (1976).

U.S. Army Corps of Engineers 1988 - Mixture of quantitative and qualitative measures to rate six functional values including wildlife. Criteria for diversity/productivity adapted from Golet (1976) to regions with different landscapes; criteria for waterfowl were adapted from Adamus (1983) but not regionalized. Also uses a "red flag" index to indicate the level of coordination and that laws protect important resources or features. Intended for wetland comparisons within an ecoregion, but scores are scaled so they may be compared across regions.

Table 2. Classes of evaluation criteria and their frequency of use in 14 wetland evaluation methods. Adamus (1983) and Adamus et al. (1987) methods contain identical classes of criteria and are combined in this table.

Wetland Evaluation Method	Diversity	Area	Land-scape pattern	Rarity	Import-ance to Wildlife	Produc-tivity	Repre-senta-tiveness	Natural-ness	Ecolog-ical Integrity	Ecolog-ical Fragility	Replace-ability	Threat	Total
Adamus 1983, Adamus et al. 1987	1	1	1		1	1							5
Ammann and Stone 1991	1	1	1	1				1	1				6
Bond et al 1992				1	1		1		1		1		5
Cable et al 1989	1	1		1									3
Durham et al 1988	1	1	1					1					4
Golet 1976	1	1	1			1							4
Gosselink and Lee 1989	1	1	1				1	1	1	1			7
Hollands and McGee 1986	1	1	1										3
Larson 1976		1		1	1		1						4
Municipality of Anchorage 1991	1	1	1	1	1	1	1				1		8
NCDEHNR[1] 1995	1	1	1	1	1			1	1	1			8
OMNREC[2] 1984	1	1	1	1	1	1	1				1		8
USACOE[3] 1988	1	1	1	1	1	1							6
Total	11	12	10	8	7	5	5	4	4	2	3	0	

[1] NCDEHNR is the North Carolina Department of Environment, Health, and Natural Resources
[2] OMNREC is the Ontario Ministry of Natural Resources and Environment Canada
[3] USACOE is the U.S. Army Corps of Engineers

Table 3. Popularity poll on criteria from reviews of conservation and wetland evaluation methods. n=number of methods in each review. Adamus (1983) and Adamus et al. (1987) methods are considered as one wetland evaluation method because they contain identical classes of criteria.

Criterion	Review of Evaluation Methods				
	Margules and Usher 1981 (n=9)	Margules 1981 (n=8)	Smith and Theberge 1986 (n=13)	Wetland evaluation methods (n=13)	Total (n=43)
Diversity (habitats and/or species)	8	8	12	11	39
Rarity (habitats and/or species)	7	6	13	8	34
Uniqueness (included with rarity)	2				2
Area	6	5	5	12	28
Landscape pattern defined as:					
Position in ecological/geog unit (Contiguity & Continuity)	1	1	1	10	13
Buffer, boundaries, shape (Fragmentation & Isolation)			6		6
Naturalness	7	6	3	4	20
Ecological Integrity (sustainability, intact processes)				4	4
Representativeness	2	5	6	5	18
Typicalness (included with representativeness)	2	1			3
Population size, abundance, and Importance for wildlife		4	2	7	13
Productivity			3	5	8
Ecological fragility	1	1	6	2	10
Threat of human interference (included with fragility)	6	2			8
Replaceability defined as:					
Restoration potential and wildlife reservoir potential	2			3	5
Potential value and accessibility	2	1	1		4

Table 4. Definition of global ranks that generally qualify a species for consideration as endangered or threatened subject to assessment of the degree of threat (after Natural Heritage Data Center Network 1993).

Rank	No. of Occurrences in the World	No. of Individuals	Degree of Threat
G1	< 6	< 1,000	species are critically imperiled throughout their range
G2	6 - 20	1,000 - 2,999	species are imperiled throughout their range
G3	21 -100	3,000 - 10,000	species are vulnerable throughout their range

Table 5. Indicators of landscape structure and function for bottomland hardwood ecosystems (after Gossolink and Lee 1989).

Indicator	Definition	Standards	Data Sources
Fraction of bottomland hardwoods remaining	Bottomland hardwoods remaining as percent of historical potential	More than 50% forest remaining	National Wetlands Inventory; aerial photography; SCS soil surveys; Federal Emergency Management Act
Bottomland hardwoods patch size & distribution	Size frequency distribution of bottomland hardwood patches	log normal distribution of patches	Same as above
Contiguity of steam to bottomland hardwoods	Length of bottomland hardwoods-stream interface divided by twice the length of the stream	ratio approaching 1	Same as above
Contiguity of upland forest to bottomland hardwoods	Length of bottomland hardwoods-upland forest interface divided by total bottomland hardwoods-upland interface	ratio approaching 1	Same as above
Water quality	Historical change in flow adjusted concentration of phosphorus	.05 mg/l for lakes; .10 mg/l flowing water	U.S. Geological Survey
Nutrient loading	Total nutrient input divided by water flux	<0.12 g/m^3 safe; 0.12-.22 g/m^3 borderline; >0.22 g/m^3 dangerous;	NPDES permits; Land use coefficients X area; Corps of Engineers discharge records
Stage-discharge relations	Historical changes in stage-discharge rating curve		Corps of Engineers discharge records; U.S. Geological Survey discharge records
Water detention	Volume of water stored on floodplain divided by discharge at flood stage		Area and contour of floodplain from topographic maps; discharge records
Balanced indigenous populations	Old growth stands; total bottomland hardwoods; endangered/threatened species; presence/absence of indicator species; change in bird species richness		State Fish and Wildlife records; Fish and Wildlife breeding bird surveys; Audubon Society Christmas bird counts

Table 6. The major categories and potential components of a population viability analysis (after Gilpin and Soule 1986).

Category	Component
Population dynamics	Number of individuals
	Age / Size distribution
	Geographic structure
	Growth rate
	Variation in demographic parameters
Population characteristics	Morphology (e.g., variation in sizes or patterns)
	Physiology (e.g., metabolism, reproduction, disease resistance)
	Behavior (e.g., breeding, interspecific interactions)
	Dispersal pattern, migration, habitat selection
Environment	Habitat quantity
	Habitat quality (e.g., abundance or density of resources and interacting species)
	Pattern of disturbance (e.g., duration, frequency, severity, and scale)

Table 7. Classification of ecological criteria for wetland protection (after Margules and Usher 1981).

Limited to site attributes, but may still be quite variable over time	Depend on extensive survey in surrounding biogeographic region	Require case studies on other similar sites and can not be assessed by a single site visit.
Area	Landscape pattern	Ecological Integrity
Diversity	Naturalness	Fragility
Productivity	Representativeness	Replaceability
Importance to wildlife	Rarity	Threat
Threat?	Importance to wildlife	
	Threat?	

Table 8. Resource categories and mitigation planning goals (from USFWS 1981).

Resource category	Designation criteria	Mitigation planning goal
1	High value for evaluation species and unique and irreplaceable.	No loss of existing habitat value.
2	High value for evaluation species and scarce or becoming scare.	No net loss of in-kind habitat value.
3	High to medium value for evaluation species and abundant.	No net loss of habitat value while minimizing loss of in-kind habitat value.
4	Medium to low value for evaluation species.	Minimize loss of habitat value.

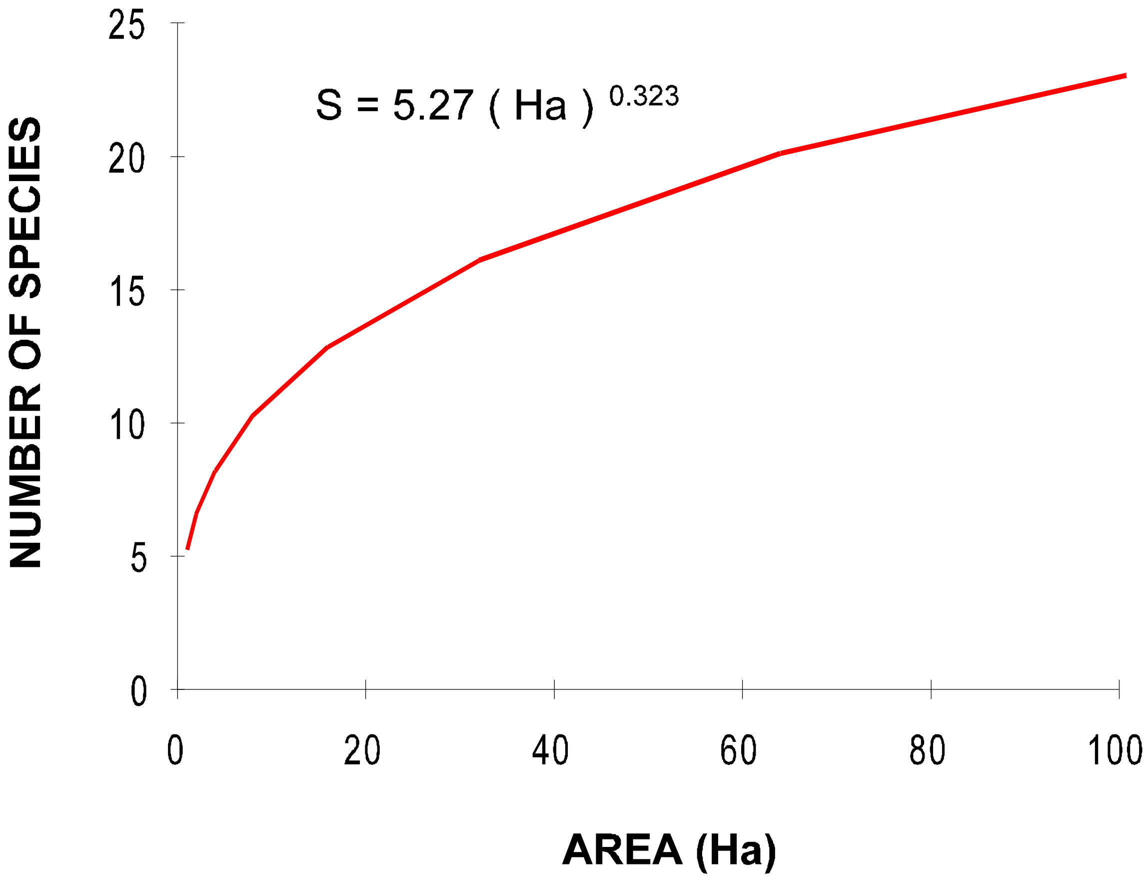

Figure 1. Wetland bird species-area relationship of Wisconsin cattail marsh (from tabulated data in Tyser [1983]).

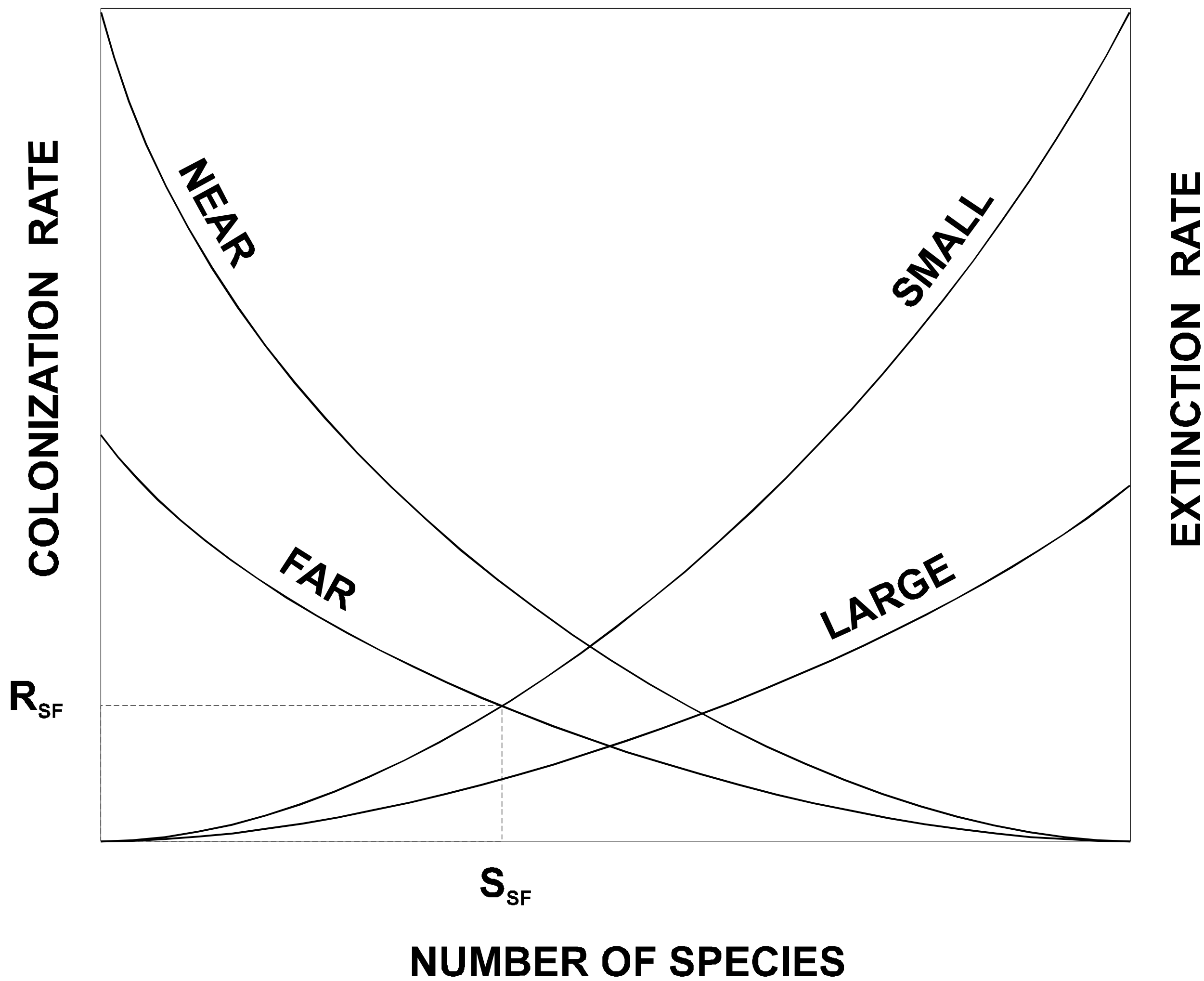

Figure 2. Extinction and colonization rates as a function of island size and isolation. R_{SF} shows the extintion and immigration rates of a small far island, and S_{SF} indicates the resultant equilibrium number of species.

Geographic Distribution		Wide		Narrow	
Habitat Specificity		Broad	Restricted	Broad	Restricted
Local Population Size	Some-where Large	Truely common	Most common form of rarity and generally as common as their habitats	Although rare, high ecologic plasticity provides good chance for introduction	Classic endemics
	Every-where Small	Although sparse, unlikely risks extinction	Chance of introduction poor and likely use all avail-able habitat	May never occur	Truely rare

Figure 3. Forms of rarity and their consequences relating to description of a species range, habitat specificity, and density (see Rabinowitz et al. 1986).

	Category	Example
physiognomic	**Class**	**Woodlands**
	Subclass	**Mainly Evergreen Woodlands**
	Group	**Evergreen Needle-Leaved Woodlands**
	Formation	**Evergreen Coniferous Woodlands with Rounded Crowns**
floristic	**Cover Type**	***Juniperus occidentalis***
	Community Type	***Juniperus occidentalis / Artemisia tridentata*** (the codominant species of the plant community by canopy layer)

Figure 4. United Nations Educational, Scientific and Cultural Organization vegetation classification format and example as adopted for Gap Analysis (from Jennings 1993).